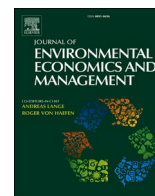


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Estimating the value of threatened species abundance dynamics

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ABSTRACT

Conservation spending aimed at helping threatened species lacks information on the marginal benefits of increases in the abundance of threatened species that occur at different points in time. This paper develops an empirical approach combining a choice experiment and a structural model to estimate two key parameters in a dynamic willingness-to-pay function: the current marginal benefit of increases in threatened species abundance and the rate implicitly used to discount future marginal benefits. An application to a threatened Coho salmon along the Oregon coast illustrates the method. We find that the public values a one-year marginal increase in Coho abundance of 1000 fish from \$0.08 to \$0.19 per household with a discount rate for future increments in salmon abundance of 2.1%. We apply these results to an instantaneous and permanent marginal increase in salmon abundance of 0.79% resulting from a policy change in one watershed and show this marginal change can generate over \$63 million in present value of social marginal benefits to the greater Pacific Northwest region. Results provide direct evidence that conservation activities that achieve immediate abundance gains for a threatened species (or prevent immediate losses) produce significantly higher benefits than activities that gradually achieve the same abundance gains.

1. Introduction

Decisions about conservation investments are classic economic choices that frequently occur at the margin: protecting a parcel of natural land from conversion to other uses through an easement; restoring a tidal wetland by removing dikes that hold back sea water; extending a regulatory no-harvest zone in riparian forests. Optimal conservation theory suggests that such investments should occur when the marginal benefits of a conservation action exceed the marginal costs (Wu and Boggess 1999; Wätzold and Drechsler, 2005; Polasky et al., 2014; de Vries and Hanley, 2015). Applying economic conservation theory to an investment decision that incrementally increases the abundance of a threatened or endangered species is challenging for many reasons, including scientific uncertainty over how a particular conservation action might impact species' abundance, and because quantifying the benefits of a conservation investment requires measurement of non-use values (Boyle and Bishop 1987; Loomis and White 1996; Richardson and Loomis 2009).

Stated preference (SP) methods provide a way to measure non-use values, though the existing SP literature is not well-suited to estimating such values arising from a marginal change in the abundance of a threatened species for two reasons. First, most studies focus on estimating the value of a change in the official conservation status of a species, such as an improvement from endangered or threatened to recovered (i.e., de-listing) under the U.S. Endangered Species Act (ESA) (Lew 2015). Recovered status is usually reached

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through a non-marginal increase in abundance. It is generally not possible to consistently derive a per-unit marginal value of *abundance* from a non-market value estimate of an improvement in recovery status. Second, the public's time preferences matter since conservation actions today may generate species abundance changes that occur at different points in time (Fig. 1, Panel B). For example, preserving an area of forest may prevent the population of a resident species from declining in the short run, while restoring adjacent forest may gradually increase abundance of the same species over a longer time period as the trees grow. The few SP studies that include abundance separately from recovered status in the experimental design typically only provide enough information to value marginal changes in abundance that occur within a restricted timeframe, for example by the end of the design's time horizon (e.g., Lewis et al., 2010) (Fig. 1, Panel A).

This paper introduces a method for estimating the present value of public non-consumptive benefits generated by efforts to increase threatened or endangered species abundance over time. Public non-consumptive benefits include non-use values and use values associated with non-consumptive activities like wildlife viewing. The method links an innovation in choice experiment (CE) design with a two-stage econometric procedure. The CE survey experimental design presents respondents with graphical and numerical information describing alternative abundance time paths for a threatened species. The econometric analysis uses conventional estimates of household willingness-to-pay (WTP) for time paths of increased abundance (first stage) in calculations that recover the structural parameters of a household-level non-consumptive dynamic WTP function (second stage). The resulting non-consumptive dynamic WTP function can then be applied to value abundance time paths that are not included in the experimental design, including marginal changes that do not result in official species recovery.

We illustrate the method using a recent choice experiment on conservation activities aimed at increasing the abundance of a threatened Pacific Coho salmon species (*Oncorhynchus kisutch*) within the federally defined Oregon Coast (OC) Coho salmon evolutionary significant unit in the United States (Lewis et al., 2019). Applying our method to this choice experiment produces estimates for two key parameters that are not identified in the prior Lewis et al. (2019) application of this data. First, estimates indicate that the public discounts future abundance changes for Coho salmon at a rate of 2.1%. Second, respondents with a 4-year college degree are willing to pay approximately \$0.19/household for a current-period one-year increase in salmon abundance of 1000 fish, which represents a marginal 0.67% increase from the current baseline abundance; respondents without a 4-year degree are willing to pay just under \$0.08/household for the same increase. An important feature of our method is that the discount rate is identified from respondents' observed trade-off between higher payments and additional salmon abundance earlier in time. Thus, our discount rate estimate explicitly captures how respondents weigh earlier versus later salmon abundance changes.

By estimating parameters in a dynamic WTP function, our method is particularly useful in monetizing a range of empirically-estimated biophysical impacts of marginal conservation investments on threatened species abundance. To illustrate the potential

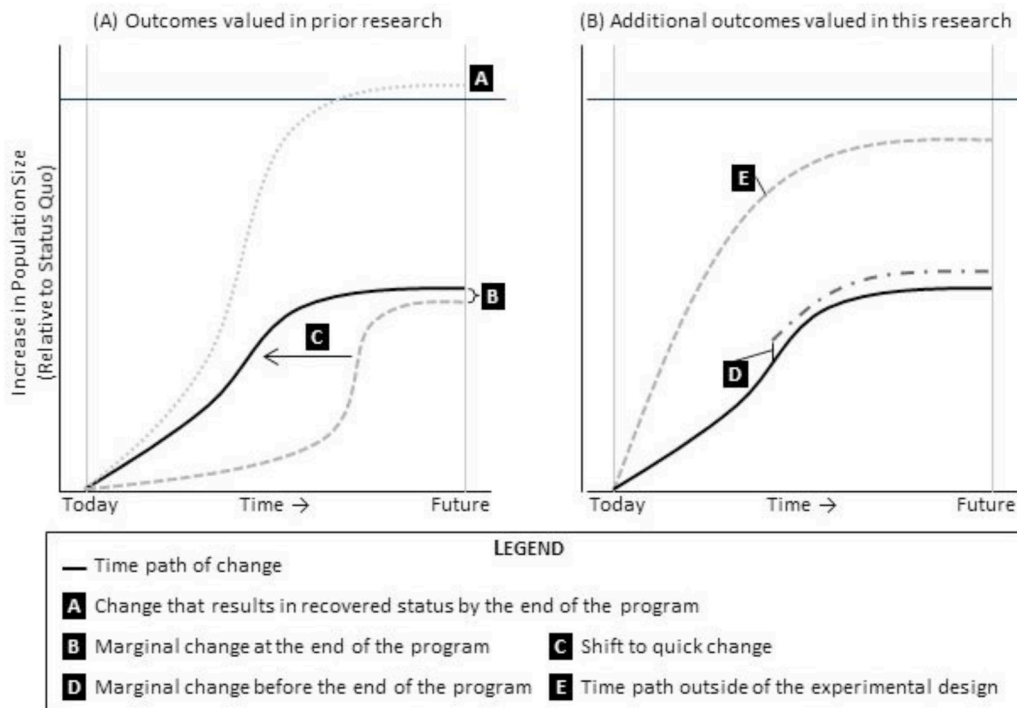


Fig. 1. Qualitative comparison of contributions made by this paper versus the existing literature. (A) Outcomes valued in prior nonmarket valuation research. (B) Additional outcomes valued in this research. Key contributions of the methodology developed in this paper include the ability to value marginal changes and associated time-paths of abundance that occur before the end of the program (Panel B, Feature D) and full time paths not included in the experimental design (Panel B, Feature E).

applicability of our estimates, we consider a recent policy change that eliminated a salmon hatchery in one watershed in our study area and which has been estimated by fisheries biologists to generate an immediate marginal increase in OC Coho salmon abundance of 1190 fish per year ($\sim 0.79\%$ of baseline abundance) (Jones et al., 2018). Applying our estimated dynamic WTP function to this 1190 fish increase in salmon abundance and scaling to the full population of just under 10 million households in our study region, we find estimated benefits of approximately \$63 million in present value. In contrast, valuing the same policy change using the standard choice experiment approach from Lewis et al. (2019) is restricted to using the time-paths of salmon abundance within the experimental design of the survey. That process significantly underestimates the benefits of the policy by 32%–49% compared to the estimates we obtain in this analysis. This highlights a key advantage of estimating parameters in a dynamic WTP function by providing the flexibility to value a range of species abundance time paths that might emerge from a variety of conservation investments (Fig. 1, Panel B), and such flexibility has been identified as desirable for benefit transfer functions (Newbold et al., 2018).

Our main contribution is a practical method that translates SP survey results to an empirical structural equation for the household-level present value of threatened species abundance changes. The results we obtain rely on the inclusion of an attribute capturing the explicit time paths of the abundance changes, which to our knowledge is absent in other CE-based species-specific valuation studies to date. With this information in the experimental design, the per-unit value of species abundance can be disentangled from household time preferences.

Our method bridges what is ultimately an unnecessary gap between the non-market valuation literature and applied research that needs non-market value estimates to compare with market-valued benefits and costs. The natural science literature is increasingly estimating empirical production functions that could be used as the basis for implementing optimal conservation theory. For example, the fisheries science literature has conducted numerous empirical assessments of the impacts of marginal conservation activities on salmon abundance, such as land use (Bradford et al., 2000), beaver dams (Leidholt et al., 1992), wood placement in streams (Roni and Quinn 2001), and boulder weir placement (Roni et al., 2006). However, while prior valuation research on Pacific salmon has found non-market benefits from discrete programs for salmon habitat enhancement (Bell et al., 2002; Garber-Yonts et al., 2004) including dam removal (Loomis 1996), these studies do not contain information needed to estimate a dynamic WTP function. Further, natural resource economics analyses of optimal conservation timing for threatened species require a dynamic WTP function, including a discount rate (Conrad 2018).

Our method also contributes to the natural capital valuation toolkit (Guerry et al., 2015). Recent work on natural capital valuation utilizes techniques from capital theory to empirically price natural stocks, which do not in general follow socially optimal time paths (Fenichel and Abbott 2014). These exercises risk substantially miscalculating the shadow price of stocks that produce economically-significant non-market benefits for the public (Yun et al., 2017). Fenichel and Abbott (2014) stress "... the vital importance of non-market valuation techniques [...] for quantifying ..." net benefits – which they label dividends – of natural capital (2014, p. 8), and call for the methodology to be extended to incorporate empirical estimates of non-market benefits (p. 19). The technique we propose makes this integration possible, and is the first, to the best of our knowledge, that offers precisely the empirical structural representation of household-level non-market values that is necessary to comprehensively value time paths of species abundance in a natural capital framework.

2. A model of willingness-to-pay for time paths of species abundance

2.1. Non-consumptive values of threatened species abundance

There is significant evidence that people hold non-market values for recovering threatened species (Richardson and Loomis 2009) and for protecting biodiversity (Jacobsen et al., 2012; Hanley and Perrings 2019). An important feature of the value for threatened species is its non-use aspect – where individuals are willing to pay to conserve species they may not ever use, either passively (e.g., wildlife watching) or consumptively (e.g., hunting or fishing). For species like salmon – the focus of our analysis – non-market values may include anglers' consumptive values from sport fishing as well as non-consumptive values from both anglers and non-anglers.¹ Non-consumptive values can consist of traditional non-use values as well as passive use values from popular activities like watching spawning salmon in the wild.² Additional species abundance may generate utility for people due to higher use values and due to the fact that people who hold non-use values may gain utility from moving the species further from extinction.

While prior SP studies have emphasized valuing conservation status changes such as "threatened" to "recovered" (Lew 2015), there is evidence that people also value marginal changes in the abundance of threatened and endangered species that have non-consumptive values. In a choice experiment analysis of the non-market benefits of protecting threatened Stellar Sea Lions found along the north coastline of the Pacific Ocean, Lew et al. (2010) included both official conservation status (endangered, threatened, recovered) and sea lion abundance as attributes in the choice cards and found evidence that respondents were willing to pay more for both additional abundance and for improved conservation status. In an earlier analysis of the Oregon Coast Coho salmon choice experiment data used in this analysis, Lewis et al. (2019) included separate attributes describing conservation status (threatened or

¹ In our sample, approximately 9% of respondents had fished for salmon in our study region within the year prior to the survey being administered, while 44% indicated at least some prior angling for Pacific salmon or steelhead in the past.

² A good example of passive use values is Oregon's Salmon Watch program, which brings thousands of elementary school children on annual field trips to find spawning salmon and learn about water quality. The species in our study – Oregon Coast Coho salmon – is a primary target species in the Salmon Watch program.

recovered), abundance (number of fish), and the speed of abundance increases (quick or slow) as attributes. Lewis et al. found evidence that respondents held positive non-consumptive values for additional abundance, earlier gains in abundance, and improved conservation status. They also found evidence of an interaction in the utility of additional abundance and conservation status, with the marginal utility of abundance diminishing as conservation status improved. It remains an open question whether the marginal utility from threatened species abundance changes is larger for species with non-consumptive use values than for species with only non-use values.

The policy relevance of positive non-market benefits for marginal changes in threatened species abundance is important. If people value marginal abundance changes, then even small conservation actions that only partially help threatened species can have economic benefits. Further, if people have a clear time preference for early abundance gains, then conservation actions that achieve higher and earlier abundance gains will generate higher non-market benefits than actions that achieve lower and slower abundance gains. Thus, any economically optimal conservation action involving threatened species requires information on the magnitude and time preferences of the people whose utility is affected by improving species abundance.

2.2. Non-consumptive dynamic WTP function

In this section, we develop a simple dynamic model of a household’s WTP for a time path of monotonic increases in the abundance of a threatened species. Suppose conservation actions are expected to increase the abundance of a species (X_t) from an initial baseline \bar{X} to a final level X_T by terminal time T , and hold abundance at that level permanently. We assume the time path of the abundance increase in time t relative to \bar{X} is described by the “beta growth function” (Yin et al., 2003):

$$\Delta abund(t, \bar{X}, X_T, T, \omega) = (X_T - \bar{X}) \left(1 + \left[\frac{T-t}{T-\omega} \right] \right) \left(\frac{t}{T} \right)^{\frac{T}{T-\omega}} \tag{1}$$

Here ω is a parameter that influences the timing of the maximum rate of change in abundance. For any given baseline and final abundance levels, lower values of ω imply a quicker path to the final abundance level X_T (Fig. 2). Therefore, a lower value of ω could be used to reflect greater conservation investments in the early years of a program. We use the beta function to represent growth because i) it allows us to fix terminal abundance (X_T) and baseline abundance (\bar{X}) while varying the rate of abundance change in our choice experiment design, and ii) provides a typical “s-shaped” growth function that is commonly used with many renewable resources like fish stocks and tree growth.³

We take a simple approach that defines b as the current money-metric benefit per unit of abundance increase, and r as the individual’s constant discount rate. Therefore, a household’s present value of willingness-to-pay (PVWTP) for a conservation program that generates a time path leading to a permanent gain in abundance is:

$$PVWTP(\bar{X}, X_T, r, T, \omega) = \int_{t=0}^T b \cdot \Delta abund(t, \bar{X}, X_T, T, \omega) \cdot e^{-rt} \cdot dt + \frac{b \cdot (X_T - \bar{X}) \cdot e^{-rT}}{r} \tag{2}$$

We label Eq. (2) the “non-consumptive dynamic WTP function”. Consider two time paths defined by Eq. (2) that differ only in terms of the value of ω : e.g., ω_1 and ω_2 , where $\omega_1 < \omega_2$ such as in Fig. 2. A convenient result is that the ratio of the PVWTP corresponding to the two time paths does not depend on b , X_T or \bar{X} :

$$\frac{PVWTP(\bar{X}, X_T, r, T, \omega_1)}{PVWTP(\bar{X}, X_T, r, T, \omega_2)} = \frac{\left\{ \int_{t=0}^T e^{-rt} \cdot \left(1 + \left[\frac{T-t}{T-\omega_1} \right] \right) \left(\frac{t}{T} \right)^{\frac{T}{T-\omega_1}} + e^{-rT} \frac{1}{r} \right\}}{\left\{ \int_{t=0}^T e^{-rt} \cdot \left(1 + \left[\frac{T-t}{T-\omega_2} \right] \right) \left(\frac{t}{T} \right)^{\frac{T}{T-\omega_2}} + e^{-rT} \frac{1}{r} \right\}} \equiv G(\omega_1, \omega_2, r, T) \tag{3}$$

The practical use of this result is the following. If estimates of PVWTP corresponding to known values of ω_1 , ω_2 , and T are available, then we can numerically solve Eq. (3) for r .⁴ A key feature of a CE survey used for this method is that values of ω and T are determined by an experimental design. Next, since Eq. (2) is linear in b , we can substitute in the estimate \hat{r} to obtain an estimate of \hat{b} , again using an estimate of PVWTP for the associated path. While there are other growth functions besides the beta function from Eq. (1) that could potentially be used, our current implementation requires that growth functions must have the property from Eq. (3) to uniquely identify the discount rate. The most straightforward implication of the non-consumptive dynamic WTP function in Eq. (2) is that individuals are willing to pay more for a time path that achieves earlier than latter gains in abundance, ceteris paribus. The implication of the ratio in Eq. (3) is that the percentage that individuals are willing to pay for a quicker time path (ω_1) compared to a slower time path (ω_2) – where $\omega_1 < \omega_2$ – is defined by the discount rate and not the baseline or ending abundance level.

³ Another key feature of the beta function is that it assumes monotonic growth, which may not be applicable in all cases.

⁴ This approach does not allow us to decompose r further to identify information about time preferences, for example household average pure rate of time preference. In future work, it may be feasible to combine a choice experiment featuring dynamic population time paths with supplementary questions designed to elicit more information about time preferences (Drupp et al., 2018; Venmans and Groom 2021).

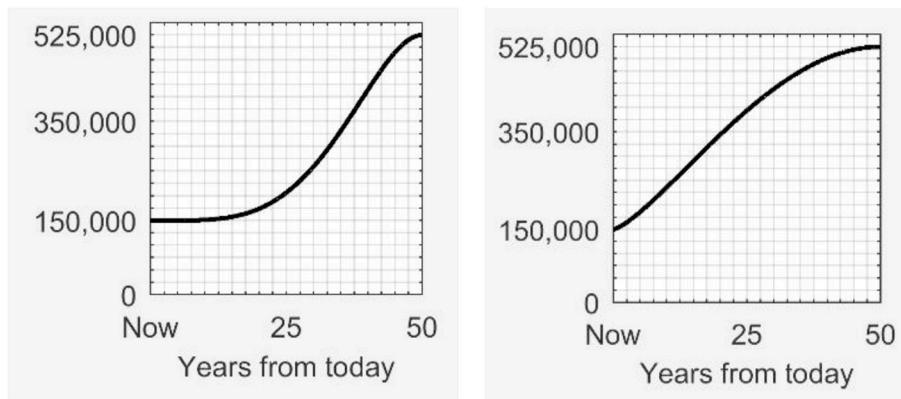


Fig. 2. Graphical depiction of the beta function with alternative ω parameters. The y-axis is the abundance of the species, while the x-axis is time. In this figure, baseline abundance (\bar{X}) = 150,000, final abundance (X_T) = 525,000, $\Delta_{abund}(T) = 375,000$ and the terminal time $T = 50$ years. These images are taken from a survey choice card.

3. Choice experiment and econometric framework

3.1. The Oregon Coast Coho Salmon choice experiment

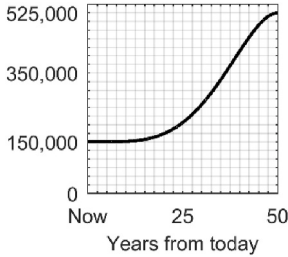
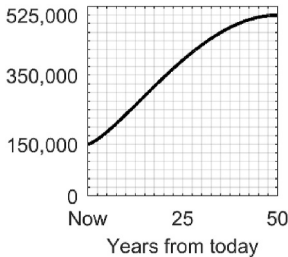
We use a choice experiment originally described in Lewis et al. (2019) to elicit preferences from the general public for conservation scenarios where the abundance of a threatened Pacific Coho salmon species is defined by a fixed baseline \bar{X} and varies by a final steady-state abundance X_T achieved in $T = 50$ years, and the speed with which abundance increases occur (i.e., the level of ω). In fall of 2017, 5000 mail surveys were sent to a random sample of households in the greater Pacific Northwest of the U.S. – Washington, Idaho, Oregon, and northern California (Bay Area and north). The choice experiment survey was designed with a set of realistic conservation investment scenarios consistent with salmon life histories and management policies to estimate the non-market values for Oregon Coast Coho salmon (*Oncorhynchus kisutch*). The survey design used two government recovery plans to guide scenario development for the surveys: the Oregon Coast Coho Conservation Plan for the State of Oregon (ODFW 2007) and the Federal Government’s 2016 Recovery Plan for Oregon Coast Coho Salmon Evolutionary Significant Unit (NMFS 2016). Moreover, input on the information provided in the survey was supplied by an advisory panel composed of fisheries biologists, fishery and resource managers, and economists. The survey was also pre-tested through several focus groups and a formal pilot survey (Lewis et al., 2019).

Survey respondents were asked their preferences across recovery scenarios that varied with the following attributes (Table 1): i) whether OC Coho were “Recovered” or remained “Threatened” under the ESA, ii) the average abundance (number) of adult fish returning to freshwater streams in a defined terminal time of 50 years, iii) the speed at which the returning adult fish population is increased over 50 years, and iv) recreational fishing regulations including fishing season and harvest limits. Each choice scenario presents respondents with a graph of the dynamic path of salmon abundance (Table 1), with differences between dynamic paths described to respondents in the survey prior to the choice scenarios (Appendix Fig. A1). An important feature of our experimental design was the use of the beta function from Eq. (1). The design randomly varies the speed of increase in salmon abundance as either “Quick” ($\omega = 13.7$) or “Slow” ($\omega = 38.7$) as in Fig. 2. So, survey respondents are explicitly asked to express their preference for a “quick” increase in salmon abundance. Given their novelty in choice experiments, we ran focus groups that probed how respondents interpret graphs depicting the dynamic paths of species abundance changes. Input from these focus groups led to the development and inclusion of a full survey page that explains how to read the graphs (Appendix Fig. A1).

We use a D_0 -optimal experimental design (Huber and Zwerina 1996) to determine attribute levels accounting for multiple correlations and restrictions among the attributes, and each survey includes three choice experiment questions – the same status quo alternative is included in every choice question with a \$0 cost, along with two conservation scenarios with non-zero costs selected from Table 1.⁵ For each of the three choice experiment questions, respondents selected one preferred choice, giving us three choice responses per survey respondent. There was a total of 20 unique survey versions randomly distributed among survey respondents. Fig. 3 presents an example choice card. The choice cards are preceded by a brief “cheap talk” script (Cummings and Taylor 1999) reminding respondents to consider their own budget before answering the choice questions. The choice exercise was also framed as a consequential choice by linking the survey to the management agency tasked with recovering the species. In particular, all respondents were told that the survey was funded by the National Oceanic and Atmospheric Administration (NOAA), and were specifically told that

⁵ We did not apply a strict D_0 -efficiency criterion in the experimental design selection process. Instead, we evaluated the top 20 most D_0 -efficient experimental designs in terms of bias and efficiency in Monte Carlo experiments (1000 iterations each) with alternative assumed true preference parameters. The selected design was able to generate parameter estimates which were significantly different from zero (5% level) on at least 99% of simulations for all parameters.

Table 1
Experimental design attributes and levels of attributes used in the stated preference survey for recovery of Oregon Coast Coho salmon given to US Pacific Northwest residents.

Attribute	Levels of attribute
Endangered Species Act Status	(1) Threatened (2) Endangered
Abundance of returning fish in 50 years	(1) 150,000 (status quo only) (2) 250,000 (3) 325,000 (4) 375,000 (5) 525,000
Speed of increase in abundance of returning fish	(1) No change (status quo only) (2) Slow (accompanied with a graph like this)  (3) Quick (accompanied with a graph like this) 
Fishing regulations (season & harvest limits)	(1) Open occasionally with 5 fish/year bag limit (2) Open annually with 5 fish/year bag limit (3) Open annually with 10 fish/year bag limit
Annual cost of program	(1) \$0 (status quo only) (2) \$10 (3) \$20 (4) \$50 (5) \$75 (6) \$100 (7) \$150 (8) \$250 (9) \$350

“NOAA is a U.S. government agency charged with managing the recovery of Oregon Coast Coho salmon”, and “Your responses will provide important information about public opinion to decision-makers tasked with helping to recover Oregon Coast Coho Salmon.”⁶

The response rate from the mail survey was 21%, accounting for deceased respondents and undeliverable surveys. An initial analysis of the choice experiment data was conducted by Lewis et al. (2019) and results indicate population-level non-market benefits of up to \$518 million for $\Delta abund = 100,000$ returning salmon in $T = 50$, and benefits of up to \$277 million for achieving conservation goals quickly ($\omega = 13.7$) rather than slowly ($\omega = 38.7$) for conservation scenarios ranging from $\Delta abund = 100,000$ returning salmon to $\Delta abund = 375,000$ returning salmon. Since the baseline salmon abundance is $\bar{X} = 150,000$ fish, the benefit estimates from Lewis et al. can be interpreted as estimates from a set of non-marginal conservation scenarios, as they do not exploit the features of the beta function of abundance growth to estimate either the discount rate r nor the current marginal benefit b of abundance. Therefore, their

⁶ While the survey questions were framed as a consequential choice, we overlooked the inclusion of follow-up questions on how respondents viewed the consequentiality of the task.

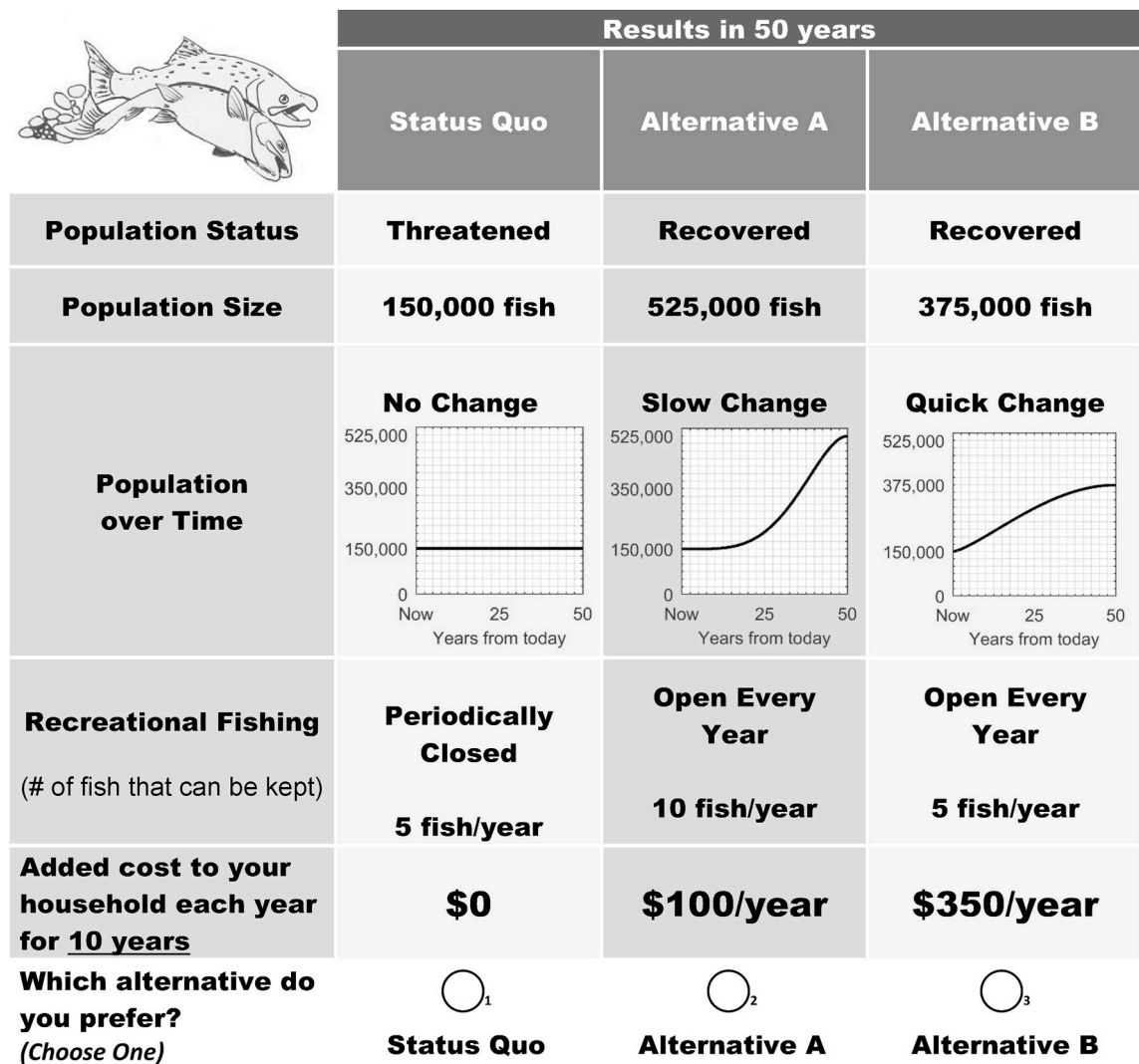


Fig. 3. Example choice card question. Attributes are selected from the experimental design in Table 1, and there are 60 unique choice cards.

model estimates are not conducive to recovering the benefits from marginal conservation investments, nor for time-paths of abundance changes that deviate from what was used in the experimental design (see Fig. 1). In the next section, we develop a two-stage econometric model to overcome these shortcomings. Interested readers are referred to additional details about survey development, experimental design testing, other pretesting activities, and survey implementation provided in an extensive supplemental document from Lewis et al. (2019).⁷ The supplemental document also provides a summary of responses to a question asked in the survey about the speed of population change, with the majority (about 58%) indicating support for a quick population change, even if it costs more money, rather than a slower change.

3.2. First-stage random utility model

We specify a random utility model that accommodates heterogeneous preferences through random parameters. This is the first-stage model that yields the estimates of WTP needed in the second stage to recover estimates of the dynamic parameters. Respondent n 's random utility U_{nik} from choosing restoration alternative i from choice question k is:

$$U_{nik} = -\exp(\beta_{n1})Price_{nik} + \beta_{n2}Rec_{nik} + \beta_4^{Quick_{nik}} \beta_5^{Rec_{nik}} [\beta_{n3} + \gamma_n College_n] \Delta Abund_{50nik} + \beta_{n6} Fishing1_{nik} + \beta_{n7} Fishing2_{nik} + \epsilon_{nik}. \tag{4}$$

⁷ Available at <https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0220260>.

Here, $Price_{nik}$ is the annual price of the restoration program (in \$100s); Rec_{nik} is a binary indicator of whether Coho salmon are officially de-listed, or recovered, under the ESA (=1) or not (=0); $\Delta Abund_{50nik}$ represents the change in abundance (relative to the baseline) of returning OC Coho salmon that occurs in $T = 50$ years (in 100,000s)⁸; $Quick_{nik}$ is a binary indicator of whether returning salmon abundance rises quickly (=1) or slowly (=0); $Fishing1_{nik}$ is a binary indicator of whether the Coho salmon fishing season is annual with a 5 fish/year limit (=1) or not (=0); and $Fishing2_{nik}$ is a binary indicator of whether the Coho salmon fishing season is annual with a 10 fish/year limit (=1) or not (=0).⁹ The status quo is modeled with an alternative-specific constant, β_{n0} representing the utility of the current state. The probability that respondent n makes choice i on choice card k is:

$$P_{nik} = Prob[U_{nik} > U_{njik} \forall j \neq i] \tag{5}$$

The specification in Eq. (4) is informed by our analysis of the representativeness of the sample to the broader population in the Pacific Northwest. We find that the sample income is representative of the population, but our sample is more likely to be i) a college graduate, ii) older than age 65, iii) male, and iv) white than the broader population in the region (Appendix Table A1). An analysis of observable preference heterogeneity in the Appendix finds that respondents with a 4-year college degree ($College_n = 1$) have a higher marginal utility for $\Delta Abund_{50nik}$, and so we include a dummy-variable indicating whether respondent n has a 4-year degree ($College_n$) as an interaction with $\Delta Abund_{50nik}$ in Eq. (4). However, we find no evidence of observable preference heterogeneity based on race, gender, or age of the respondent in our supplemental analysis in the appendix. Further, we use methods from Cameron and DeShazo (2013) and Kolstoe and Cameron (2017) and find no evidence of sample selection based on unobservables (Appendix). Therefore, we only include $College_n$ in Eq. (4) to indicate observable heterogeneity. Unobservable preference heterogeneity is captured through random parameters in Eq. (4), except for the multiplicative parameters β_4 and β_5 which enters as fixed.¹⁰

The random utility specification can be used to estimate the present value WTP for a conservation program leading to abundance increases that are defined by the terminal time abundance change ($\Delta Abund_{50}$) and the dummy variable indicating speed of abundance change ($Quick$) as:

$$WTP(\Delta Abund_{50}, Rec, Quick, College) = \frac{\beta_2 Rec + \beta_4^{Quick} \beta_5^{Rec} [(\beta_3 + \gamma College) \Delta Abund_{50}]}{\exp(\beta_1)} \tag{6}$$

For the case where “recovery” is not achieved ($Rec = 0$), the ratio of WTP of interest for determining the non-consumptive benefit function discount rate in the second stage reduces to¹¹:

$$\frac{WTP(\Delta Abund_{50}, Rec = 0, Quick = 1, College)}{WTP(\Delta Abund_{50}, Rec = 0, Quick = 0, College)} = \beta_4 \tag{7}$$

For our purposes, the fact that the WTP ratio in Eq. (7) reduces to a single estimated parameter is a key advantage of the multiplicative RUM specification in Eq. (4) since it allows us to identify the respondent discount rate r with a single RUM parameter by application of the theory in Eq. (3). A key restriction for our dynamic WTP function is the fact that we are restricting the discount rate to be constant over a potentially long time-horizon. While we adopt this assumption to simplify identification of the discount rate to a single parameter, we acknowledge that future research could attempt to relax this assumption.

3.3. Second-stage calculation of non-consumptive benefit function parameters

Given first-stage estimates of random utility model parameters from Eq. (4), calculating non-consumptive benefit function parameters b and r in the second stage is fairly straightforward. First, an important consideration is that, since WTP for a scenario depends on college attainment (Eq. (6)), second stage computation of b will also depend on college attainment. This means we must calculate two estimates of b : $\hat{b}_{No\ College}$ for non-college households, and $\hat{b}_{College}$ for households with college attainment.

To generate point estimate (median) values of \hat{r} , $\hat{b}_{No\ College}$, and $\hat{b}_{College}$ along with confidence intervals, we adapt standard Krinsky and Robb (1986) simulation methods to the second-stage calculations. We first draw J sets of RUM model parameters ($\tilde{\beta}_j$) using the point estimates and fitted covariance matrix from the first stage. For each j^{th} parameter vector draw, we simulate a distribution corresponding to each random parameter using H Halton draws. Since each simulated $\tilde{\beta}_{aj}$ is a fixed rather than heterogeneous

⁸ The connection between first stage variables $\Delta abund_{50nik}$ and $Quick_{nik}$ in Eq. (4) and the beta growth function $\Delta abund(t, \bar{X}, X_T, T, \omega)$ (Eq. (1)) is as follows. The experimental design uses the same value of baseline abundance (\bar{X}) and terminal time ($T = 50$) in all choice occasions. Only ω and X_T vary. The value of $Quick_{nik}$ sets ω to one of two values (quick or slow). Conditional on this ω_{nik} and the final absolute abundance $X_{T,nik}$, $\Delta abund_{50nik} = \Delta abund(T, \bar{X}, X_{T,nik}, T, \omega_{nik}) = X_{T,nik} - \bar{X}$.

⁹ The recreational fishing regulation change variables were included to empirically test for the presence of consumptive option values that could arise from increased angling opportunities with a higher salmon stock, and from non-consumptive values associated with bequest motivations regarding future angling opportunities.

¹⁰ We were unable to obtain convergence when β_4 and β_5 were specified as random, likely due to the multiplicative nature of these parameters.

¹¹ The experimental design defines the payment vehicle as “[a]dded cost to your household each year for 10 years”. We rely on this definition to write the ratio in Eq. (6) as involving WTP rather than PVWTP as shown in Eq. (3), since the sum of constant discrete discount factor weights applied to the numerator and denominator to evaluate the present value computation would cancel, leaving the (constant) annual WTP terms.

parameter, we find a single value of \tilde{r}_j by setting Eq. (3) equal to $\tilde{\beta}_{4j}$ and solving numerically. Our preferred point estimate, \hat{r} is the median value of \tilde{r}_j , and we produce a confidence interval by calculating quantiles of the simulated discount rates.

To obtain $\hat{b}_{\text{No College}}$, and \hat{b}_{College} , we simulate distributions for these parameters utilizing values of \tilde{r}_j and $\tilde{\beta}_j$. We first use \tilde{r}_j along with the j^{th} parameter vector draw to produce draws of the present value of WTP for different time paths of abundance in the experimental design that do not lead to recovery ($PV\tilde{WTP}_j(\text{College})$).¹² Next, we use Eq. (2) to calculate values of $\tilde{b}_{\text{No College},j}$ and $\tilde{b}_{\text{College},j}$ corresponding to \tilde{r}_j . Similar to how we handle \tilde{r}_j , we calculate median values and confidence intervals from the simulated distributions of $\tilde{b}_{\text{No College},j}$ and $\tilde{b}_{\text{College},j}$.

4. Estimation results

Parameters from the random parameters logit (RPL) model in Eq. (4) are estimated with maximum simulated likelihood using 2000 Halton draws and results are presented in Table 2. Parameter estimates indicate strongly significant and positive coefficients for the main utility parameters ($p < 0.05$) except for the two dummy variables regarding recreational fishing regulations (fishing 1, fishing 2), which are jointly insignificant from zero using a likelihood ratio test ($p < 0.05$). The joint insignificance of the recreational fishing variables indicate that our non-market value estimates are likely dominated by non-consumptive values associated with increased salmon abundance.¹³ The other parameters indicate that respondents gain utility from official recovery (de-listing), from higher terminal period abundance of salmon, and from achieving terminal period abundance levels quickly. We also find evidence of strong respondent heterogeneity in preferences, including that college graduates gain more utility than non-college graduates for steady-state salmon abundance.¹⁴

Parameter estimates for the second-stage dynamic parameters are presented in Table 3. The estimated $G(\omega_1, \omega_2, r, T)$ ratio from Eq. (3) is found from $\hat{\beta}_4 = 1.36$, indicating that the average WTP for a quick increase in salmon abundance ($\omega = 13.7$) is 36% larger than the average WTP for a slow increase ($\omega = 38.7$) that generates the same final abundance target. Given $G(\omega_1, \omega_2, r, T) = 1.36$, the implied average rate at which respondents discount future abundance increases is solved numerically as $\hat{r} = 0.021$. Finally, the current period benefits that respondents derive for an extra 100,000 returning salmon in a given year is determined by solving for b from Eq. (2), using the estimated discount rate of $\hat{r} = 0.021$. We find that $\hat{b}_{\text{No College}} = \7.83 per 100,000 fish per household and $\hat{b}_{\text{College}} = \19.09 per 100,000 fish per household for college graduates. Expressed for a more marginal change, the per-household estimates are $\hat{b}_{\text{No College}} = \0.08 per 1000 fish and $\hat{b}_{\text{College}} = \0.19 per 1000 fish. As expected, college graduates are willing to pay more for an increment of abundance than those without a college degree.

5. Application – valuing the observed partial recovery of coho salmon in Salmon River, OR

5.1. Valuing effects from a hatchery removal

We now illustrate an application of our estimated parameters from the non-consumptive dynamic WTP function to recent conservation actions aimed at increasing the abundance of Oregon Coast Coho salmon in the Salmon River along the central Oregon coast. The Salmon River is a small watershed with approximately 81 km of Coho spawning habitat that has received multiple conservation interventions. A system of dikes in the Salmon River estuary was removed between 1978 and 1996, re-establishing historical tidal marsh acreage that can be used by Coho. To assist the recovery of Oregon Coast Coho salmon, the Oregon Department of Fish and Wildlife ceased stocking hatchery Coho into the Salmon River in 2007. A “before-after-impact-control” (BAIC) analysis¹⁵ of returning adult Coho to the Salmon River finds that the average annual abundance of fish has increased by approximately 1190 individual fish during the years 2013–2016, an effect attributed to the hatchery program discontinuation (Jones et al., 2018). Importantly, the hatchery discontinuation led to “an immediate wild population increase during the post-hatchery period” (Jones et al., 2018 p.51). An

¹² Specifically, if \tilde{WTP}_j is the j^{th} Krinsky-Robb (KR) simulation of median household annual WTP, then we form $PV\tilde{WTP}_j = \tilde{WTP}_j \sum_{t=0}^{10} \left(\frac{1}{1+r_j}\right)^t$. Since

first-stage WTP results used in this calculation involve functions of random parameters, we take the median over the H draws from the random parameters corresponding to the j^{th} KR simulation iteration.

¹³ The prior analysis of this data in Lewis et al. (2019) performed an attribute non-attendance test on the fishing regulation dummy variables and estimated a probability of non-attendance for these variables of 93% using the Hess et al. (2013) latent class-RPL approach. Attribute non-attendance was much higher for the fishing regulation dummies than for the other attributes and was the only one with a probability of non-attendance greater than 50%.

¹⁴ The prior analysis of Lewis et al. (2019) uses the same variables that we do in Eq. (4), but with a different specification where all variables were linear and additive, e.g., the *Quick* variable was included as a linear term and as an interaction with $\Delta Abund$. While our specification in Eq. (4) has a slightly worse fit (log-likelihood of -2020.59) than the linear and additive specification from the Lewis et al. (2019) paper (log-likelihood of -2016.16), our specification has the advantage of imposing the restrictions required to estimate the dynamic WTP parameters by imposing the $G(\omega_1, \omega_2, r, T)$ ratio from Eq. (3). We refer interested readers to additional supplemental analyses in Lewis et al. (2019), who found that results were robust to potential protest respondents, attribute non-attendance, and stratified sampling.

¹⁵ This is conceptually similar to the difference-in-difference method which is a more familiar term for economists.

Table 2
Random parameters logit estimates of choice model.

		Main Model	
		Parameter	Std. Error
Status Quo	ASC	-11.81*	1.70
Parameter Means	log (Price)	0.66*	0.12
	Recovered	1.26*	0.37
	$\Delta Abund_{50}$	0.58*	0.19
	Quick interaction	1.36*	0.07
	Recovered interaction	0.63*	0.11
	Fishing1	-0.08	0.12
	Fishing2	0.07	0.20
Parameter standard Deviations	SD (Price)	1.74*	0.10
	SD (Recovered)	2.00*	0.39
	SD ($\Delta Abund_{50}$)	0.41	0.50
	SD (Fishing1)	0.48	0.41
	SD (Fishing2)	0.25	0.87
	SD (ASC)	10.00*	1.43
College interaction	College interaction	0.65*	0.21
	SD (College interaction)	0.06	0.51
	Log-likelihood	-2020.59	

Note: * indicates $p < 0.05$.

Table 3
Estimates of dynamic WTP parameters.

	\hat{r} (discount rate)	$\hat{b}_{No\ College}$ (per 100,000 fish gained)	$\hat{b}_{College}$ (per 100,000 fish gained)
Estimate (median) (95% CI)	0.021 (.0132, 0.0274)	\$7.83 (\$1.47, \$20.20)	\$19.09 (\$8.71, \$34.29)

extra 1190 fish per year is a marginal change (0.79%) in the ESU-wide abundance.

Valuing the benefits of 1190 additional returning adult salmon provides an example of the usefulness of our non-consumptive dynamic WTP estimates as a means of transferring benefits. A key feature in an “idealized benefit transfer function” is to estimate an appropriate unit value (Newbold et al., 2018), and the unit in this exercise is a specific change in salmon abundance (+1190 salmon) over a specific period-of-time (immediate). An immediate increase of 1190 additional fish is worth approximately \$0.141/household/y using our structural parameter estimates.¹⁶ Multiplying across the 9,408,059 households in the greater Pacific Northwest, the benefits are \$1,327,917/y. The study by Jones et al. (2018) was an ex-post evaluation that did not assess whether the additional abundance of fish would be expected to increase more. If we interpret the increased fish to the Salmon River as an immediate change following Jones et al.’s (2018) interpretation, then the present value of an infinite stream of 1190 additional fish/y is \$63.2 million when evaluated at our estimated 2.1% discount rate.

5.2. Valuing dynamic paths – alternative paths to achieve a 1190-fish steady-state increase

We highlight the role of dynamics in threatened species abundance by exploring the magnitude of benefit differences that arise from imposing alternative dynamic paths to the terminal period increase in returning salmon abundance of 1190 fish from the previous section. Consider three paths of dynamic increases in fish abundance, where all three result in a permanent increase of 1190 returning fish after the end of 50 years (Fig. 4, Panel A). The slow and quick paths are defined using parameters from our experimental design, including the terminal time T and rate of change parameter ω . We compare the benefits of these paths to the benefits from an instantaneous increase described in section 5. a, which is outside of our experimental design and which we can only value because of our estimation of the non-consumptive dynamic WTP function. This exercise allows us to examine the flexibility benefits that arise from having dynamic WTP function parameters in comparison to the standard choice experiment approach from the same data found in Lewis et al. (2019) and illustrates the usefulness of a dynamic WTP function for benefit transfer.

While all three time paths that we consider ultimately produce the same gain in abundance, the non-consumptive value they deliver differs significantly. The root of this difference is evident when one compares the present value of abundance gains realized through each year (Fig. 4, Panel B). The instantaneous increase produces the most non-consumptive benefit early on when the penalty imposed by discounting is smallest. This same feature explains the separation in the present value of gains realized by the quick vs. the slow path through year 50.

Our model of household-level non-consumptive benefits assumes that households value future abundance levels after year 50, when the population is stable regardless of the time path. Inspecting the contribution of overall PVWTP from gains prior to year 50 and

¹⁶ This is a population weighted measure reflecting that 35.8% of the population has a 4-year college degree.

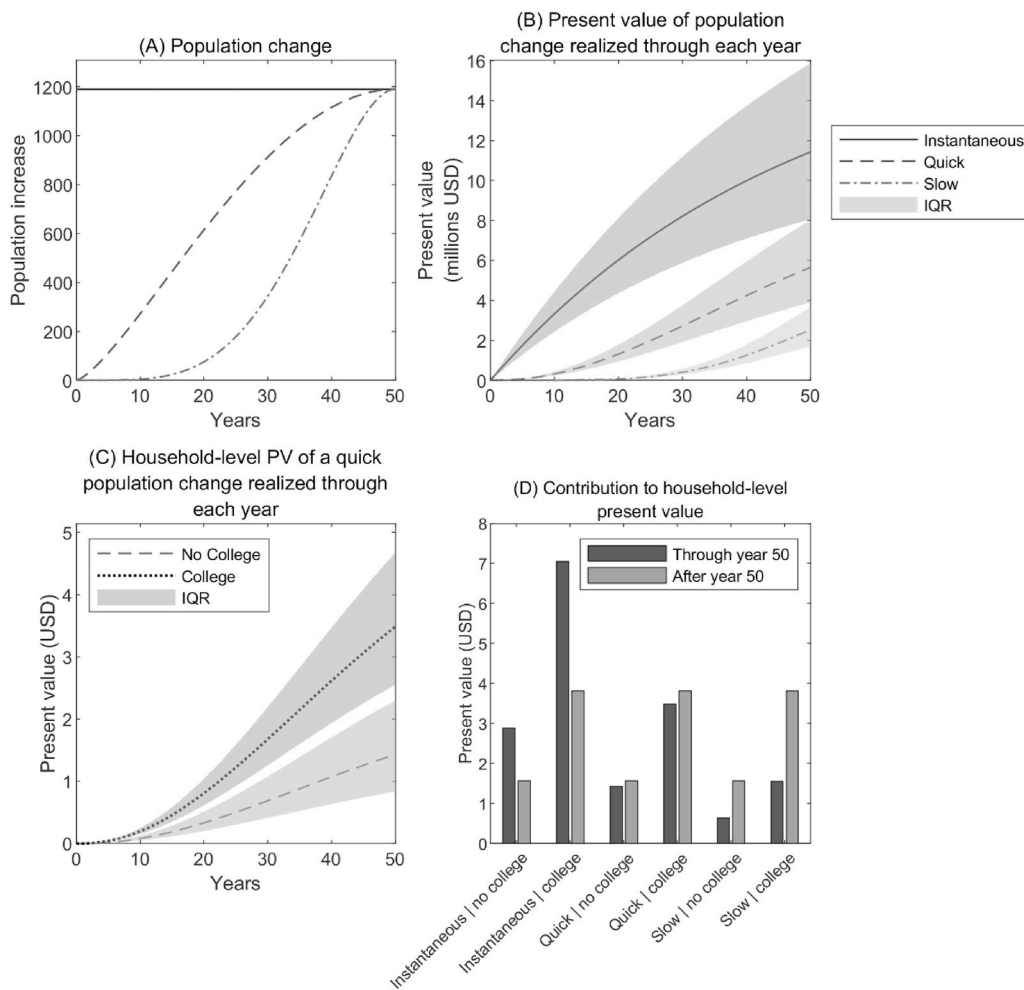


Fig. 4. Comparing the value of time paths of abundance gains for a threatened species. (A) Time paths of population increase. (B) Population-level present value of WTP for time paths shown in Panel A, shown in terms of PVWTP for population gain realized through each year. (C) Household-level PVWTP for the quick time path (Panel A). (D) Contribution to overall household PVWTP from each time path pre- and post-year 50.

after year 50 reveals that non-consumptive benefits from the future after year 50 can comprise the majority of the value unless the time path increases to near the final value rapidly (Fig. 4, Panel D). This result is driven partly by the low estimated range of the discount rate, r (95% CI: (0.0132, 0.0274)). Since we find that households with college attainment have a higher current non-consumptive value per unit of abundance, they accrue greater benefit from early gains in abundance (Fig. 4, Panel C). However, they also receive a larger benefit from permanent gains farther in the future relative to non-college households, all else being equal, and so the relative value of time paths pre- and post-year 50 are qualitatively similar (Panel D). A further implication is that efforts aimed at either restoring or saving an immediate change in salmon abundance by amount $\Delta Abund$ provide significantly more benefits than restoration efforts that would gradually increase salmon abundance to eventually reach $\Delta Abund$. In particular, the present value population benefits of a permanent instantaneous increase in 1190 returning salmon is approximately \$63 million, while the corresponding present value benefits of a slow (quick) increase that eventually reaches 1190 returning salmon after 50 years is just over \$32 million (\$43 million).

This comparison of the present value of benefits over a range of time paths that reach the same final species abundance highlights the need for valuing abundance time paths outside of the experimental design used in our survey. Eliminating the hatchery led to an estimated immediate increase in returning salmon, and imposing one of the paths from our experimental design – slow ($\omega = 38.7$) or quick ($\omega = 13.7$) with a terminal time of $T = 50$ – would have led to a significant underestimate of the non-consumptive benefits of between 32% and 49%. Therefore, the advantage of estimating parameters in the non-consumptive dynamic WTP function compared to the standard choice experiment approach such as used in Lewis et al. (2019), is the ability to value a wide range of scenarios involving abundance changes over time.

6. Discussion

Many individual conservation investments are motivated by the plight of threatened and endangered species, and the accumulation

of numerous marginal investments comprises many official species recovery plans. Valuing marginal investments in natural capital to improve threatened species habitat faces at least two key challenges. First, the benefits from improving the abundance of threatened and endangered species include non-use values (Krutilla 1967), and measurement of such values typically requires stated preference methods (Freeman 2003). Second, conservation investments generate time-paths of marginal changes in species abundance, and valuing the dynamics of species abundance changes requires knowledge of parameters in a non-consumptive dynamic WTP function: the current period benefit of an increment in abundance and the rate with which people discount future abundance change values. While there is a significant stated preference literature on threatened and endangered species (e.g., see reviews by Richardson and Loomis 2009; Lew 2015; Hanley and Perrings 2019), this literature has focused on valuing official recovery status changes (e.g., ESA listing status) and has not generated key dynamic parameters in a WTP function that could be used to value marginal changes in threatened species abundance. Therefore, the existing non-market valuation literature generally is unable to value individual natural capital investments (Fenichel and Abbott 2014) that marginally increase species abundance but do not lead to species recovery, and is unable to put monetary values on ecological evidence of the impacts of marginal conservation investments on threatened species abundance (e.g., Roni and Quinn 2001; Jones et al., 2018).

This paper develops an approach to recover key dynamic WTP parameters for changes in the abundance of a threatened species, and we apply the method to the problem of estimating the benefits of incremental conservation efforts aimed at increasing the abundance of a Coho salmon species (*Oncorhynchus kisutch*) within the federally defined Oregon Coast (OC) Coho salmon evolutionary significant unit, which is listed as threatened under the ESA. The choice experiment method requires an experimental design where i) respondents are presented with final period species abundance levels which are varied in the design, and ii) respondents are presented with some depiction of the time-path of abundance gains (e.g., a graph) which result from application of the biological beta function with known parameters which are varied in the design (e.g., quick vs. slow). Given these basic experimental design elements embedded in a choice experiment, we develop a two-stage estimation process to generate dynamic WTP parameters, with i) a first-stage random utility model with multiplicative parameters that can be estimated with conventional random parameters approaches, and ii) a second-stage where a WTP ratio is used to implicitly solve for average respondent discount rates which then identify the current period per-unit benefit of an increment in species abundance. The non-consumptive dynamic WTP parameters can be used to value any time path of changes in species abundance and can be used to estimate benefits for a population of individual consumers. Using empirical estimates that eliminating a salmon hatchery in a single watershed in our study region immediately increased wild Coho salmon abundance by 1190 fish, we apply our method to show that this conservation investment generated population benefits of \$1.3 million/y, and a steady-state stream of such benefits is worth a present value of \$63 million when evaluated at our estimated 2.1% discount rate for salmon abundance. We show that an alternative conservation strategy, which slowly increases salmon abundance to the same 1190 fish change, generates roughly half the level of benefits. Thus, our results provide direct empirical evidence that conservation activities that achieve immediate abundance gains for a threatened species (or prevent immediate losses) produce significantly higher benefits than activities that gradually achieve the same abundance gains. Our results also highlight the flexibility advantages that come from the ability to value any time-path of species abundance changes through estimation of dynamic WTP parameters, and so our method contributes to recent calls for flexible benefit transfer functions (Newbold et al., 2018) as well as to natural resource economic analyses of optimal conservation timing for threatened species (Conrad 2018).

Empirical estimates of conservation impacts on physical abundance of threatened species are voluminous in the natural science literature, and our approach contributes an important new application of stated preference non-market valuation estimates to put monetary values on the estimated impacts of small conservation efforts on threatened species abundance. Our approach integrates all current best practices in stated preference design (Johnston et al., 2017) with a simple methodological approach that will greatly increase the flexibility with which non-market valuation estimates can be used. However, we do not claim that our approach is the only method that could be used to estimate dynamic WTP parameters for species abundance. For example, one could use an experimental design that includes a small abundance change attribute combined with prior choice experiment approaches that use split-sample designs to estimate average financial discount rates (e.g., Kim and Haab 2009) or use investment trade-off survey questions to estimate individual discount rates (e.g., Newell and Siikamäki, 2014). One drawback to using financial investment questions or split-sample designs on payment vehicle length to estimate discount rates is the problem that there may be time-preference inconsistencies between payments and benefits (Crocker and Shogren 1993; Abdellaoui et al., 2009), which implies different discount rates for financial payoffs and future benefits. In contrast, our method estimates discount rates for benefits using respondent trade-offs between payments and the time-period where species abundance changes (benefits) occur. Whichever approach is taken to estimate dynamic WTP parameters, developing a strong future linkage between stated preference research, optimal conservation theory, and empirical natural science production function estimation could be used to inform conservation decision-making and better integrate areas within environmental economics and between environmental economics and ecology.

We view this research as advancing the state-of-the-art of integrating ecological-economic policy analysis models and non-market valuation models. Simply put, our view is that these literatures should “talk to each other” more effectively: non-market valuation studies should produce estimates that models can use, and models should generate credible scenarios that can be valued. This level of integration is rare in applied work, and we see two extensions to this research that would help to further bridge the divide. First, while the expected abundance time paths we include in the SP survey are credible in that they are within the envelope of recovery plan assumptions, we did not produce the quick and slow time paths using an empirical population-dynamic model. Building on proof-of-concept provided by this research, the next step for SP studies of species conservation should be to investigate the feasibility of incorporating output from such a model into the experimental design. The potential benefits of doing so include enhanced credibility of WTP function estimates and stronger internal consistency of policy analysis involving non-market benefits.

A second related area for further work on this methodology is understanding the impact of uncertainty on values for marginal

abundance changes and time preferences revealed by SP surveys. In our application, salmon abundance numbers exhibit substantial year-to-year variability, and while the survey instrument informs respondents of sources of variability like ocean conditions, the expected abundance paths shown in the experimental design do not include stochasticity. A key question that has not been investigated, to our knowledge, is whether the choice experiment method can support a design that includes both dynamic paths of abundance change and intuitive measures of year-to-year variability. Understanding whether non-market valuation surveys can isolate public values for both properties of species abundance changes over time is critical in order to determine the extent to which non-market values can be incorporated into policy analysis.

Declaration of competing interest

The authors declare no known conflicts of interest related to this manuscript.

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Appendix. Sample selection analysis and value aggregation to the population

Scaling up our sample value estimates requires an analysis of whether our sample is representative of the broader population of the Pacific Northwest study region (ID, WA, OR, northern CA). First, we compare demographic information from our sample to the broader population (Table A1), which shows that our sample is representative of the broader population's income but not the other demographic variable we have access to. In terms of the other demographics, our sample is more likely to be i) a college graduate, ii) older than age 65, iii) male, and iv) white than the broader population. To examine whether our key utility parameters differ by these observable demographic characteristics, we estimate the primary model with an interaction between these demographic characteristics and our key utility parameters similar to how the *College* dummy variable is interacted in Eq. (4). We then conduct likelihood ratio tests of the null hypothesis that the mean and standard deviation of the random parameter on the demographic interactions are jointly zero, with p-values for each test presented in the last column of Table A1. We reject the null that the *College* dummy variable has no interaction with the utility parameters at any reasonable significance level, but fail to reject the same null for the other demographic variables. Thus, our primary model in Eq. (4) only uses the *College* dummy variable to represent observable heterogeneity in the utility parameters.

Second, we analyze the possibility of sample selection based on unobserved characteristics by following methods developed by Cameron and DeShazo (2013) and Kolstoe and Cameron (2017) for use in logit models. This approach was used in the earlier Lewis et al. (2019) analysis of this same data, and we repeat it here with the new functional form of the random utility model. The approach first estimates the propensity to respond to the survey using a binary logit model with observable demographic information that we have for both respondents and non-respondents. In particular, we use racial and income data on the census tract where each individual lives, the percentage Democratic vote in the 2016 presidential election for the county where each individual lives, state dummies, and information about mail delivery for each individual. The mail delivery information includes whether respondents get their mail curbside, from a central location, or from neighborhood delivery and collection box units (NDCBUs). Estimation results are presented in the supplemental information in the earlier analysis of this data from Lewis et al. (2019) – see Table M in that paper's supplemental information. The presence of NDCBUs has a significant negative effect on the propensity to respond to our survey, likely because the survey packets were too big to fit into the outgoing mail slots in such units. Thus, the mail collection variables serve as the exclusion restriction since they affect the propensity to be in our survey but likely have no effect on people's utility of salmon conservation. The mean propensity to respond is 0.21 (equal to our response rate), and we compute a variable *pdiff* that represents the difference between each individual's estimated propensity to respond and the mean of 0.21. We included an interaction between *pdiff* and the key utility parameters similar to our analysis of observable demographic characteristics above, and then conduct a likelihood ratio test of the null hypothesis that the mean and standard deviation of the random parameter on the *pdiff* interactions are jointly zero. The p-value from this test is 0.85, and we thus fail to reject the null that utility parameters have no interaction with *pdiff*. Thus, there is no evidence of sample selection bias from unobservable characteristics.

Table A1

Demographic characteristics for the sample and population of Pacific Northwest (PNW) residents for the Oregon Coast Coho salmon stated preference survey. P-values are from a likelihood ratio test of the null hypothesis that demographics do not interact with utility parameters.

Demographic Statistic	Sample	Pacific Northwest Population	Is Sample Representative?	p-value for utility interaction
Household Median Income	\$60 k - \$80 k	\$70 k	Yes	–
Percent College Grads	54.0%	35.8%	No	0.006

(continued on next page)

Table A1 (continued)

Demographic Statistic	Sample	Pacific Northwest Population	Is Sample Representative?	p-value for utility interaction
Percent Age 65+	38.5%	15.1%	No	0.91
Percent Male	61.1%	49.8%	No	0.97
Percent White	87.7%	72.0%	No	0.40

Oregon Coast Coho Population over Time

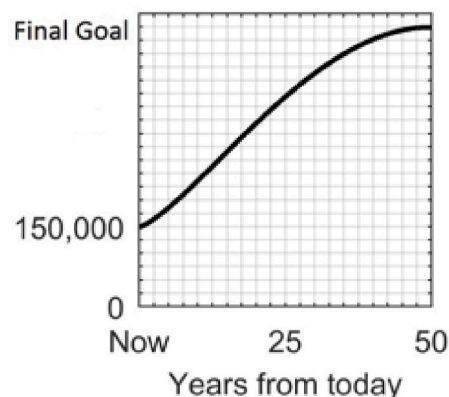
Depending on how restoration activities are done, the change in the population size of Oregon Coast Coho salmon may be **quick** or **slow**.

Quick Change

If habitat is restored in the immediate future, Coho salmon population will rise **quicker**.

A **Quick Change** could mean starting at today's population size of an average of **150,000 fish**, then increase abundance to a **Final Goal** in 50 years. There would be about a **65%** increase in the population in the first 25 years.

Example of Quick Change



Slow Change

If habitat is restored more gradually, Coho salmon population will rise **slower**.

A **Slow Change** could mean starting at today's population size of an average of **150,000 fish**, then increase abundance to a **Final Goal** in 50 years. There would be about a **15%** increase in the population in the first 25 years.

Example of Slow Change

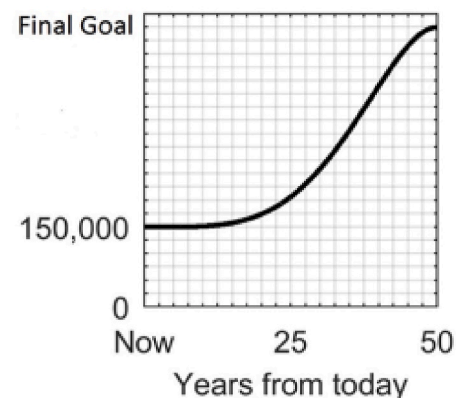


Fig. A1. Description of quick and slow change in Oregon Coast coho populations as presented in the survey.

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